Planned burning in Tasmania. I. A review of current practice and supporting information

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Abstract

Planned burning is the deliberate use of fire under specified conditions for the purposes of managing fuels, ecological systems, agricultural green pick or weeds. This paper reviews planned burning as applied in Tasmania, and the underpinning base of knowledge and experience. Subsequent papers cover the fire risk assessment procedure for planned burning, and revised guidelines for conducting planned burning in Tasmania. The aim of this work is to minimise the risk of adverse outcomes from planned burning whilst ensuring that burning is performed safely and meets fire management objectives.

Introduction

Fire is a fundamental aspect of the Australian environment, with many vegetation types requiring periodic fire to maintain ecological values. However, under certain conditions fire can threaten human life and property, may be too frequent or too intense, and can cause temporary reductions to air quality and/or disruptions to the public.

Planned burning, the deliberate use of fire under specified conditions, has the potential to address some of these issues. This paper covers low-intensity planned burning, and issues relating to fuel management, enhancing regeneration, ecological management, agricultural weed management, but not high-intensity regeneration burning in wet eucalypt forests. In the past, planned burning was normally referred to as fuel-reduction burning. This term could, however, be misleading due to the low correlation between fuel load and fire spread rate, and as fuel-reduction burning only refers to one burning objective. Hence, it is more appropriate to specify the burn's objectives, the target outcomes and the degree of fuel and ecological management required.

One stated aim of planned burning is minimising wildfire risk, in particular the risk of high-intensity wildfires. High-intensity wildfires are responsible for the majority of the threats to public health and safety, are extremely difficult and expensive to suppress, and may threaten ecological values through both their lack of fire regime variability and the small proportion of the landscape left unburnt. Planned burning can also assist with the maintenance of ecological values by providing a range of fire types, seasons, frequencies, ages, sizes and intensities.

However, planned burning is a balancing act and is not a panacea for all fire management problems, with many fire management issues being more closely aligned with social rather than operational factors, particularly near the urban interface (Kanowski *et al.* 2005). In this area, the overriding factor is ensuring that appropriate fuel management is conducted immediately adjacent to (i.e. within about 550 metres from) assets (mostly

houses, but also critical infrastructure such as urban services, electricity and telecommunications infrastructure).

Major wildfires have occurred in Tasmania. The most recent occurred in February 1967, when over a five-hour period 62 people died, approximately 1,400 buildings were destroyed and 265,000 ha burnt (Luke and McArthur 1978). However, the area of Tasmania burnt in the February 1967 fires is dwarfed by that burnt in the 1933/34 and 1897/98 fires. The 1933/34 fires burnt over 1,000,000 ha (mostly in western and southwestern Tasmania) and the 1897/98 state-wide fires burnt over 2,000,000 ha (Marsden-Smedley 1998a).

Thus, although Tasmania has avoided catastrophic wildfires for several decades, it is not immune to the threat. It is worth noting that many of the areas burnt in the 1933/34 and 1897/98 fires consisted of "bush", but by February 1967 these areas had been built up with houses, farms and forest plantations, and since February 1967 many additional areas of "bush" have been developed, resulting in potentially increased levels of damage should fires occur.

Over the past few decades there have been marked reductions in the area of planned burning on both private and crown land in Tasmania (Robson 1993; Kirkpatrick and Bridle 2007; von Platen 2008). At the broad scale, this has resulted in increases in the average age since fire (i.e. the time since the last fire), with resultant increases in the overall level of fuel hazard. As a result, enhanced application and implementation of fire management practices is required if land management agencies and fire authorities are to address this issue. This enhanced fire management will require high-level inter-agency cooperation, along with an improved understanding of the interactions between asset protection, community aspirations regarding fire, fire management planning, fire risk assessment, fire behaviour and suppression, and the ecological management of fire-prone areas.

Revised and updated guidelines for planned burning are one aspect of this improved fire management. The Tasmanian fire management agencies (the Tasmania Fire Service, Forestry Tasmania and the Parks and Wildlife Service), through the Tasmanian Fire Research Fund, therefore reviewed and updated the planned burning guidelines used in Tasmania (Marsden-Smedley 2009). Revised guidelines aim to minimise the risk of adverse outcomes from planned burning whilst also ensuring that burning is performed safely and meets fire management objectives. A critical aspect of these revised guidelines is the linking of clearly defined objectives with measurable outcomes.

Performing planned burning in Tasmania

Fire regime

A site's fire regime comprises a wide range of factors including age (i.e. time since fire), frequency (i.e. time between fires), season, patchiness and intensity (Gill 2008), with the site's management aims determining the most important factors. When performing fuel management burns, the critical issues are ensuring adequate burn coverage and fire intensity. In contrast, when managing for ecological values (e.g. species diversity), variability in fire frequency, season, patchiness and intensity may be the most important.

Bradstock *et al.* (2005; see also Gill *et al.* 2003) use the concept of the visible versus the invisible mosaic. The visible mosaic consists of the current variation in fire age, season, patchiness and intensity. The invisible mosaic, which is as important for ecological management, is the temporal variation in the frequency, season, patchiness and intensity of previous fires. Site time since fire is normally the main fire regime factor used in fire planning and operations. This is due to the ease with which time since fire can be assessed, mapped and incorporated into planning along with its influence on

vegetation structure, fuel hazard, fuel continuity and fuel load. Fire frequency and burn coverage are much harder to assess as they combine the effects of a number of fire events.

Fire management zoning

Fire management zone types and their names are determined by legal requirements and the management aims of the land manager. Some general zone types can, however, be defined, including asset zones, asset-protection zones, strategic management zones, general land management zones and planned burning exclusion zones (Department of Sustainability and Environment 2006, for example).

The asset zone covers the geographic location of defined high-value assets, such as urban areas, buildings, ecological assets and/or communication infrastructure, with zone size being dependent on the asset's characteristics. Planned burning for fuel management would not normally be conducted within the asset zone, although ecological management burning may be performed for the maintenance of rare and/or threatened fire-dependent species. In most situations, fire risk in asset zones will be managed by the manual removal of fuel hazards and by requiring appropriate building designs.

Asset-protection zones are located immediately adjacent to assets and/ or ignition sources, with the primary objective being intensive fuel management to minimise wildfire risk. In this zone, ecological values, viewfields and/or recreational opportunities are of secondary importance and may be adversely impacted. As such, the area of the asset-protection zone needs to be kept as small as practicable.

The strategic management zone aims to provide broad-scale fuel management to increase wildfire suppression potential and reduce wildfire size whilst minimising adverse impacts on other values. This means that the strategic management zone needs to be of sufficient size and continuity to act as a barrier to fire spread by reducing the rate of spread, intensity and spotting under a broad range of fire weather conditions and/ or allowing for effective fire suppression operations.

The general land management zone aims to allow for land management in keeping with the land manager's requirements. This zoning aims to maintain fire regimes for vegetation management (e.g. species and structural diversity), cultural heritage, catchment management, weed management and/or fire exclusion. This zone should provide for a range of ecological objectives and requirements for both flora and fauna.

Planned burning exclusion zones may be located within other land management zones. These areas may have vegetation types that are unsuitable for planned burning (e.g. rainforest, wet eucalypt forest), have fire-sensitive geology and/or vegetation types (e.g. karst, rainforest, alpine areas), have unsuitable site characteristics (e.g. too steep), and/or planned burning may result in unacceptable visual impacts (e.g. sites adjacent to scenic lookouts).

Effectiveness of planned burning

The effectiveness of planned burning needs to be assessed at two levels: the effectiveness of an individual planned burn and that of the planned burning process. The effectiveness of individual planned burns will need to be specified during the burn's approval process then assessed against these objectives. For a fuel management burn, the effectiveness of the planned burning process will need to be judged by the burn's potential to increase fire suppression potential and the probability that subsequent fires will self-extinguish. For an ecological management burn, the critical issues will be related to the burn's impacts and effects on target species.

Increases in fire suppression potential and the probability that fires will self-extinguish will mainly be achieved through reductions in the level of fuel hazard. Under catastrophic levels of fire danger, wildfires will typically sustain burning (although at reduced rates of fire spread and intensity) within areas that have been recently subjected to a planned burn. This means that planned burning is only one of the bushfire risk reduction strategies that needs to be implemented, and to be effective must be integrated with other strategies such as appropriate land-use planning, asset management, and control of ignition sources.

For example, recent research in the Otway Ranges using the Phoenix fire behaviour prediction system (Victorian Bushfires Royal Commission 2010b; see also Tolhurst *et al*. 2007) indicated that, although planned burning can result in major reductions in bushfire risk and the effectiveness of planned burning can be enhanced through the strategic placement of planned burning blocks, under catastrophic wildfire conditions the maximum possible reduction in level of impact is about 70%. This means that, regardless of the level of planned burning performed, there will a residual risk that has to be managed through other strategies. It also indicates that, if fuel management is not performed immediately adjacent to assets and/or ignition sources, planned burning will only provide moderate levels of fire protection.

The potential for fuel management burning to reduce wildfire spread and intensity has been documented in several Australian studies (e.g. McArthur 1962; Peet and Williamson 1968; Billing 1981; Grant and Wouters 1993; Robson 1993; Cheney 1996; McCarthy and Tolhurst 2001). The critical aspects influencing the success or failure of fuel management burning include the burn location (relative to assets being protected and/or ignition sources), size and width, coverage, proportion of the landscape treated, intensity, frequency and the weather conditions during subsequent wildfires.

The burn block size and width required for planned burns to be effective will be dependent, in part, on the vegetation type and site conditions. In dry eucalypt forest, planned burn blocks in the order of 1,500 ha with a width of at least three kilometres are recommended (Victorian Bushfires Royal Commission 2010b). The size and width of planned burn blocks required in buttongrass moorland, heathland and grassland have not been comprehensively researched but, due to their lower spot fire potential, narrower planned burn blocks should be effective at containing high-intensity bushfires in these vegetation types.

The proportion of the block actually burnt has an important influence on the effectiveness of planned burns. Where burns have a low coverage, bushfires may sustain with moderate to high rates of spread, intensity and ember numbers in the remaining fuels. Information from Western Australian dry eucalypt forests suggests that, for planned burns to be effective, burn coverage rates of greater than 60% are required, while coverage rates above 90% are unnecessary and could result in adverse ecological impacts (Victorian Bushfires Royal Commission 2010b).

The proportion of the landscape treated will influence the effectiveness of planned burning. For example, following the February 2009 bushfires in Victoria the Victorian Bushfires Royal Commission recommended that a rolling target of 5% of public land be planned burnt per year (Victorian Bushfires Royal Commission 2010a). The exact basis of this 5% recommendation is unclear but appears to be based on expert opinion (Victorian Bushfires Royal Commission 2010b). Only very limited information is available as to the proportion of the landscape that needs to be subjected to planned burning in order to reduce wildfire risk. For example, fire regime research in southwest Tasmania

suggested that, to minimise threats to rainforest and alpine areas, it was necessary to burn 10% of the highly flammable buttongrass moorlands every year. This research also found that, if the planned burning was conducted strategically, a similar level of protection could be gained by burning only 3% of the buttongrass moorland every year (King 2004a, 2004b; King *et al.* 2006, 2008). In southwestern Western Australian dry eucalypt forests, broad-scale planned burning of 8% of the forests per year is recommended, and this has been credited with reducing the potential for major wildfires (Victorian Bushfires Royal Commission 2010b).

In order to minimise both the risk of planned burns escaping and the level of resources required to perform the burn, low levels of fire intensity are often utilised during planned burns. Provided these low-intensity burns have adequate coverage rates they are normally effective at reducing the level of surface and near-surface fuels. However, in dry eucalypt forests (and especially stringybark forests), bark hazard removal is a critical objective of fuel management burning, and low-intensity fires frequently do not result in the effective removal of elevated and bark fuel hazards (Davis 2010). Hence, in order to reduce dry eucalypt forest bark fuel hazard, flame heights of at least two to four metres are required. An outcome of performing planned burns with these levels of intensity will be increased levels of scorch and potentially high levels of post-burn leaf fall, which can result in increases in surface fuel hazard after the fire.

The frequency at which planned burns are performed will influence their effectiveness for fuel management. Research in Victorian dry eucalypt forests by McCarthy and Tolhurst (2001) found strong correlations between the time since burning and the fire suppression potential, due to the influence of time since fire on the level of fuel-hazard. Burning intervals of no more than 3 years were highly effective in achieving fire suppression potential, burning intervals of 3-6 years were moderate to highly effective, and burning intervals of 10 years or more were minimally effective (due to the recovery of near-surface, elevated and bark fuel hazard).

Public education on the advantages and disadvantages of planned burning is a critical component of effective fire management. In particular, the public needs to be informed that, in order to effectively manage wildfire risk, fuel hazard management must be performed immediately adjacent to assets (especially houses) that are located within about 700 m of the urban-bushland interface (Chen and McAneney 2004). This means that, to be effective at reducing the level of wildfire threat, fuel management adjacent to assets is required regardless of land tenure (i.e. including on private land). If this fuel management is not performed, then regardless of the amount of planned burning performed on adjacent public land there will be a high level of risk to public health, safety and assets.

Fire risk assessment

Fire risk assessment can be used to identify areas with a high likelihood of being burnt along with a high consequence if fires occur. It can also be used to predict the impacts (positive and negative) of different fire management strategies (e.g. changes in the amount and location of planned burns and/or changes in resource level and location).

The Burn Risk Assessment Tool (BRAT) (Slijepcevic *et al.* 2007; Marsden-Smedley and Whight 2011) provides a standardised, objective, consistent and repeatable framework for assessing planned burn risks versus benefits. BRAT assesses the risk of escapes (i.e. likelihood of impact), potential for damage (i.e. consequence), effect of mitigation strategies in reducing escape probability, and the burn's potential to meet fire management objectives (i.e. benefits).

Burn block design

The selection of suitable burn block locations and boundaries (including any additional boundary preparation required) must be performed at an early stage in the planning process. The planned burning block shape should as much as practical avoid convoluted and/or steep boundaries. The type of boundaries used will depend on the vegetation type, terrain, presence or absence of tracks, roads, water courses and/or other low-fuel zones.

Where practical and safe, the use of non-flammable vegetation (e.g. scrub boundaries that are too wet to burn, green paddocks) as fire boundaries is the most effective strategy. Where tracks or roads are used, all boundary preparation and/or reinforcement must be completed prior to ignition. In general, larger burns may provide more effective burning conditions as these have less boundary relative to area (Forestry Tasmania 2005a, 2005b).

Some of the major factors that need to be taken into consideration when designing planned burning blocks include the relative location of assets versus hazards, location of potential ignition sources, burn block size and shape, location of suitable boundaries, fuels within and adjacent to the burn block, and special values within and/or adjacent to the burn block. When planned burns are proposed, much of the background information required is available from map databases, reports and published sources. However, ground surveys are typically required to ensure that this information is up-to-date, correct and representative of the area planned for burning.

Fuel characteristics

Prior to 15 years ago fuel characteristics meant total litter fuel load (Luke and McArthur 1978). More recently there has been a growing realisation that fire spread rate is poorly correlated with fuel load, but well correlated with fuel structure and composition (Gould 1993; Marsden-Smedley and Catchpole 1995b; Gould *et al.* 2007a). This has been addressed in fuel-hazard rating systems (e.g. McCarthy *et al.* 1999; Department for Environment and Heritage 2008; Gould *et al.* 2007a, 2007b; Hines *et al.* 2010).

When fuels are assessed, dead fuel up to 6 mm in diameter and live fuel up to 2 mm in diameter is included. All dead bark likely to be burnt in a fire is also included. In dry eucalypt forests, the height (or depth as appropriate) and cover of the surface, near-surface, elevated and bark fuels are used to predict fuel-hazard rating (Hines *et al.* 2010). In buttongrass moorlands, fuel age is used as a surrogate for fuel hazard (Marsden-Smedley and Catchpole 1995a, 1995b). In native grasslands, percentage curing is used to estimate fuel-hazard (Cheney and Sullivan 2008). In heathlands and dry scrub, fuel height is used to estimate fuel hazard (Anon 1998; Catchpole *et al.* 1998). In wet scrub, fuel height and age are used to estimate fuel hazard (Marsden-Smedley 2002).

In native vegetation, the main fuel strata are surface fuels, near-surface fuels, elevated fuels and bark fuels (Hines *et al.* 2010). The overall fuel-hazard rating is defined as the sum of the surface, near-surface, elevated and bark fuel hazard scores (Department for Environment and Heritage 2008; Hines *et al.* 2010). The main fuel factor influencing the rate of fire spread is the near-surface stratum (Gould 1993; Marsden-Smedley and Catchpole 1995b; Gould *et al.* 2007a, 2007b). The surface fuel stratum is comprised of dead grass, leaves, bark and twigs, predominantly in a horizontal orientation and in contact with or close to contact with the soil surface. Surface fuels frequently contain the majority of the fuel load and often have elevated fuel moisture contents and relatively low aeration. The near-surface fuel stratum consists of both vertical and horizontal live and dead fuels, and is made up of suspended bark, leaf litter, low shrubs, bracken, tussock grasses and sedges. In

some sites, the surface and near-surface fuel stratums intergrade with no clear break between them. Near-surface fuels are typically about 10-30 cm deep, but may be as high as 1 m. Due to their proximity to surface fuels, near-surface fuels will normally be burnt in a fire. The elevated fuel stratum consists of shrubs and tall bracken, which have a largely vertical orientation. They are typically about 1-2 m tall, but may be 8-10 m tall in wet eucalypt forests and mixed forests.

The main bark types affecting fire behaviour are smooth or gum barks, "platey" bark and stringybark. Gum bark (also known as candle bark) consist of long, coiled bark strips which may burn for extended periods and be lofted in the fire's convection column, resulting in the potential to cause long-distance spotting (i.e. greater than 2 km). Platey bark (bark that tends to form small "plates") from peppermints, ironbarks and pines is characterised by layers of dead bark which can flake off and cause short- to medium-range spotting (i.e. up to 2 km). Stringybarks form fibrous wads which can be removed by fire and result in extensive short- to medium-range spotting. Some bark types, notably stringybarks, may contribute up to seven tonnes per hectare to the fuel load (Hines *et al*. 2010), contributing to fire intensity and providing massive amounts of potential firebrand material.

Forest canopies mostly affect fire behaviour through influences on wind speed and, during high-intensity crown fires, influences on spot fire number and spotting distance.

Weather

Weather has a major influence on fire behaviour both directly and indirectly. The major weather factors directly influencing fire behaviour are wind speed and atmospheric stability. The major weather factors indirectly affecting fire behaviour through their influence on fuel moisture are relative humidity (RH), soil dryness (measured as Soil Dryness Index, SDI), wind speed, cloud type and cover, and

temperature. The major factors affecting the SDI are rainfall intensity and duration, time since rainfall, vegetation type and temperature.

The major issues related to measuring wind speed are its highly changeable nature (Gould *et al.* 2007a) and the difficulty of measuring wind speed in many sites. For correct wind speeds to be measured, large areas free of obstacles are required, with the width of the open area being at least 10 times the height of surrounding obstacles (Bureau of Meteorology 1997). Alternatively, where clearings of sufficient size are not available, the 10 m wind speed can be estimated using the Beaufort scale.

Wind speed is strongly affected by friction from the ground surface (Bureau of Meteorology 1997), which means it is also necessary to record the wind measurement height. Wind speed should be measured as the surface wind speed at 1.7-2 m above the ground surface, or as the wind speed at 10 m above the ground surface. In open sites, the wind speed at 10 m above the ground averages about 1.5 times the surface wind speed (Marsden-Smedley 1993; Tran 1999). In forested sites, Tran (1999) and Gould *et al.* (2007a) found that wind speed at 10 m above the ground averaged about 2.5 times the surface wind speed. Tran (1999) also found an approximately 50% reduction in the 10 m wind speed between open and forested sites with canopy densities of about 20-55%.

The stability of the atmosphere along with the presence or absence of inversion layers has major influences on fire behaviour (Bally 1995; Mills and McCaw 2009). This mainly relates to the likelihood that air from different altitudes will mix down to the ground surface and/or the likelihood that fires will form large convection columns. Haines (1988) developed the Haines Index in order to incorporate information on atmospheric conditions into fire management operations, combining the effects of atmospheric stability and moisture

content. The major advantages of the Haines Index are simplicity and ability to provide information from higher altitudes above the ground surface. In doing so, the Haines Index extends the fire danger rating by including weather information from above the ground surface.

However, a major issue with the Haines Index is that, with up to 25% of days in Tasmania in summer having an index of five or six, it provides poor discrimination between weather events which have high levels of atmospheric stability (this issue is more of a problem in inland mainland Australia, where during the fire season about 50 to 75% of days have an index of five or six; Mills and McCaw 2009). In order to address this problem, Mills and McCaw (2009) developed the continuous Haines Index (C-HAINES) which varies between zero and a maximum of about 13.

Precipitation includes all moisture deposited from the atmosphere, and has a strong influence on fuel moisture and hence fire dynamics. The amount and duration of precipitation along with the time since precipitation is used to predict the Soil Dryness Index (SDI) (Mount 1972). The SDI provides an estimate of the amount of rainfall required to saturate the soil, and incorporates longer-term influences on coarse fuel moisture and the flammability of different vegetation types.

The moisture content of the atmosphere is normally described by Relative Humidity (RH) and dew-point temperature. The RH is calculated as the water vapour pressure in the air relative to the saturation vapour pressure at that temperature, expressed as a percentage. A major characteristic of RH is its dependence on temperature, with warm air being able to hold a greater amount of water vapour than cold air. The dew-point temperature is the temperature at which the vapour pressure of the moisture present in the atmosphere equals the maximum vapour pressure that the atmosphere can hold (i.e. 100% RH).

Humidity influences fire behaviour through several mechanisms. RH is a major driver of fuel moisture, particularly when it falls below about 30%. At low fuel moistures, embers tend to stay alight for extended periods, resulting in increased potential for spot fires. In addition, low fuel moistures have a major influence on fire behaviour. When the dry bulb temperature falls to the dew-point temperature and forms dew, there is typically a rapid increase in fuel moisture content and a corresponding decrease in the level of fire behaviour.

Other than through its influence on the saturation vapour pressure, dry bulb temperature has minor influences on fire behaviour. Dry bulb temperature does, however, have major influences on fire crew fatigue and the risk of dehydration, and thus on the ability of fire crews to manage fires and perform fire management operations.

Fuel moisture

For fire management purposes, the term fuel moisture is defined as the moisture content of fine dead fuel (of diameter less than 6 mm) and calculated as the weight of water relative to the fuel's oven-dry weight, expressed as a percentage. Fuel moisture during planned burning can be determined using moisture meters and/or estimated using hazard sticks, prediction models or the SDI.

In dry eucalypt forests, fires will frequently fail to sustain burning when dead fuel moisture exceeds about 20-25%, but will typically burn with excessive intensities and with a high risk of spot fires when the dead fuel moisture is less than about 11-13% (Tolhurst and Cheney 1999). In buttongrass moorlands, fires will fail to burn with adequate intensity and/or continuity when dead fuel moisture exceeds about 35%, but will typically burn with excessive intensity when dead fuel moisture is less than about 15% (Marsden-Smedley and Catchpole 1995b, 2001).

Fuel moisture in wet and dry eucalypt forest and in wet scrub can be estimated using the change in weight of hazard sticks (Eron 1991; Forestry Tasmania 2005a, 2005b; JB Marsden-Smedley, unpub. data). Hazard sticks are made using *Pinus radiata* arrays which have a diameter of 12 mm and dry weight of 100 g. Hazard sticks are placed within the fuels to be burnt and in surrounding vegetation, and integrate current and recent past conditions. The main disadvantages of hazard sticks are that the relationship between stick moisture and fuel moisture is vegetation-specific, so sticks require standardisation time in the field (typically 10 to 14 days) prior to estimates of fuel moisture being made, and stick life is typically less than about 12 weeks (Eron 1991).

Fuel moisture models, using easily measured environmental parameters such as temperature, RH, dew-point temperature, wind speed, solar radiation and/or recent rainfall, can be used to predict fuel moisture. These models have the advantage of being able to make predictions using remotely collected data (e.g. data from automatic weather stations), but also have the major disadvantage that the models are vegetation-specific and should only be used within the bounds of the data used to develop the model. In dry eucalypt forest, the Matthews fuel moisture model (Matthews 2006; Matthews *et al.* 2010) is recommended. In all other Tasmanian vegetation types, the buttongrass moorland fuel moisture model is recommended (Marsden-Smedley 1998b; Marsden-Smedley and Catchpole 2001); this uses precipitation in the previous 48 hr and the current RH and dew point temperature.

The other major tool used in Tasmania to estimate fuel moisture is the SDI, which is used to predict the relative flammability of different vegetation types, and fuel removal during planned burns. When buttongrass moorland burns occur at an SDI below 10, wet scrub boundaries will be too wet to burn and will form safe fire control lines. Similarly, wet gullies in dry forest may fail

to sustain burning when the SDI is below about 25. The SDI also strongly influences the fuel moisture profile, with fuels under low SDI conditions (i.e. less than 10 in buttongrass moorlands, and less than 25 in dry forests) typically showing a strong gradient in surface-fuels moisture between the moist lower fuels and drier upper fuels. This means that, for planned burning to be effective for fuel management, at least moderate SDI levels are required (i.e. in buttongrass moorland an SDI between 5 and 20, and in dry forests an SDI greater than 50). In contrast, during ecological management burns the aim may be to leave significant amounts of unburnt fuel; this can be achieved by burning with a low SDI (i.e. in buttongrass moorland a SDI between five and 10, and in dry forests a SDI between 25 and 50).

Slope

The relationship between slope and fire behaviour was examined by McArthur (1967; see also Noble *et al.* 1980), who predicted that the rate of fire spread would double for every 10° of slope uphill and halve for every 10° of slope downhill. While McArthur (1967) provides no application bounds for this relationship, it has been suggested (K Tolhurst, pers. comm.) that the relationship should not be used on slopes outside the range of -10° to +20°.

The slope correction factor developed by McArthur (1967) assumes that fires are travelling straight up or down the slope. In many cases, this will not be the situation with many fires being burning across a slope. The available data for Tasmanian vegetation suggests that the slope in the direction of fire travel provides realistic corrections of fire spread rates (JB Marsden-Smedley, unpub. data).

Fire behaviour

To date in Tasmania, fire behaviour prediction systems have been developed for Tasmanian buttongrass moorlands

Table 1 Fire behaviour prediction systems recommended for use in Tasmanian vegetation associations.

Vegetation association	Fire prediction system
Dry eucalypt forest and woodland	McArthur Forest Fire Danger Meter (McArthur 1967) Project Vesta (Gould *et al.* 2007a, 2007b)
Buttongrass moorland	Buttongrass moorland fire prediction model (Marsden-Smedley *et al.* 1999)
Heathland, dry scrub	Heathland fire model (Anon 1998; Catchpole *et al.* 1998, 1999)
Wet scrub	Scrub Fire Danger prediction system (Marsden-Smedley 2002)
Native grasslands	CSIRO grassland fire prediction model (Cheney *et al.* 1993)
Flammable weeds	Scrub Fire Danger prediction system (Marsden-Smedley 2002)

(Marsden-Smedley *et al.* 1999) and heathlands (Anon 1998; Catchpole *et al.* 1998; Marsden-Smedley 2002). For dry eucalypt forest the McArthur Forest Fire Danger model (McArthur 1973) has been the standard system for over 30 years, with the Vesta Fire Model being recently developed (Gould *et al.* 2007a, 2007b). While predictions

of grassland fires are not routinely made in Tasmania, the Northern Territory grassland model (Cheney *et al.* 1993) has been used. When predictions of fire behaviour are made in Tasmanian vegetation types, the fire behaviour prediction systems detailed in Table 1 are recommended.

The main factors influencing fire behaviour are wind speed, fuel characteristics and fuel moisture, with wind speed being the dominant factor (Sullivan 2009). However, the relative importance of these factors varies at different wind speeds. At low to moderate wind speeds (i.e. < 25 km/h), wind speed and fuel characteristics have similar levels of influence on fire behaviour in buttongrass moorlands and dry eucalypt forests (Marsden-Smedley 1998b; Gould *et al.* 2007a). At higher wind speeds (i.e. > 25 km/h), wind speed becomes the dominant influence on fire behaviour (Marsden-Smedley and Catchpole 1995b). The rate of fire spread is estimated from its quasi-steady state, which is the fire's average spread rate once minor variation from short-term changes in wind speed, fuel characteristics and/or topography has been accounted for.

Fireline intensity is normally described using Byram's Intensity (Byram 1959), which is a function of fuel energy content, fuel load and the rate of fire spread. Fireline intensity can be used to predict flame height, with relationships being available for dry eucalypt forests, heathlands and

Photo 1. Tree scorch after fuel management burning.

Table 2. Fire danger rating system used in Tasmania (from AMEC-National Bushfire Warnings Taskforce, 2009).

Fire danger rating		Difficulty of control (suppression)
Class	Rating	
Low	0 to 5	Fire control relatively easy
Moderate	6 to 11	Direct attack on fires possible if well resourced
High	12 to 24	Fire control difficult and frequently fails
Very high	25 to 49	Fire control very difficult
Severe	50 to 74	Fire control unlikely to be feasible or safe
Extreme	75 to 99	Fire control not feasible or safe
Catastrophic	100+	Very high threats to life and property

buttongrass moorlands (Marsden-Smedley and Catchpole 1995b; Anon 1998; Catchpole *et al.* 1998; Gould *et al.* 2007a, 2007b).

Following ignition at a point, fires go through an acceleration phase, with the fireline length required for fires to achieve their quasi-steady state being dependent on vegetation type and wind speed (Cheney and Gould 1995). This length varies from 50-100 m for buttongrass moorland fires burning with wind speeds of up to about 30 km/h (Marsden-Smedley and Catchpole 1995b), to about 100 m for grasslands, and up to about 300-450 m for forest fires burning with high wind speeds (Gould *et al.* 2007b).

The tree scorch height is mainly a function of fire intensity (i.e. flame height and Byram's Intensity), temperature and wind speed. Scorch is mainly a concern in fuel management burning in dry eucalypt forest (Photo 1), due to its potential to increase litter fall and/or result in tree damage, reduced growth rates and/or death. Scorch height typically averages six to eight times flame height in spring, but 10-14 times flame height in autumn due to the typically drier fuels then (Australian Capital Territory 2008; Department of Sustainability and Environment 2008). Where trees less than about 10-15 m tall occur within planned burning blocks (e.g. in heathlands, buttongrass moorlands and dry eucalypt woodlands), it is normally not possible to prevent scorching (and frequently torching) as the entire canopy is within the flame zone.

The issue of whether fires will sustain or self-extinguish (i.e. go out without fire suppression or the use of boundaries) is of critical importance to fire management in general, and planned burning specifically. Planned burns may be undertaken in sites without internal boundaries and/or with an aim to only burn part of a site, resulting in the requirement to predict when fires will self-extinguish. Systems examining these thresholds have been developed for buttongrass moorlands (Marsden-Smedley *et al.* 2001) and native grasslands (Leonard 2009). In buttongrass moorlands, the main factors influencing the likelihood of fires sustaining are wind speed, fuel moisture and site productivity (Marsden-Smedley *et al.* 2001). In native grasslands, the main factors influencing the likelihood of fires sustaining are fuel moisture, fuel load and wind speed (Leonard 2009).

The primary aim of a Fire Danger Rating (FDR) is to provide a description of fire suppression difficulty (see Luke and McArthur 1978). In Tasmania, three systems are used for estimating fire danger: Forest Fire Danger Rating (FFDR, McArthur 1973), Scrub Fire Danger Rating (SFDR, Marsden-Smedley 2002), and Moorland Fire Danger Rating (MFDR, Marsden-Smedley *et al.* 1999). Each FDR integrates the influences

Photo 2. Fire-control line in use during planned burning.

of fuel, site factors and weather into a dimensionless index of fire behaviour and difficulty of control (suppression). The Tasmanian fire danger rating system has recently been updated (AMEC-National Bushfire Warnings Taskforce 2009), and consists of a rating class and numerical value which varies between 0 (fires will not sustain) up to in excess of 100 (Table 2).

Planned burning operations

The methodology used to ignite planned burns (e.g. ignition spacing, fireline length ignited, orientation to the slope and/or wind direction) will influence the fire's rate of spread, intensity, spotting potential and control options. Planned burn maps can be used to indicate key features of the burn (Figure 1).

The length of active fireline has major influences on the rate of fire spread, fire intensity and spotting potential. For example, grassland fires with fireline lengths of about 10 m, 25 m, 50 m, or greater than

100 m, will burn with about 40%, 75%, 90% or 100% respectively of their potential fire spread rates (Cheney and Gould 1995). This means that, during planned burns, fire behaviour can be reduced by keeping the fireline length short.

Fire behaviour is also strongly influenced by the orientation of the ignition line to the direction of the slope and/or wind direction, with fires burning as head fires, flank fires or back fires. For example, in buttongrass moorlands, flank and back fires typically burn with about 40% and 10% respectively of the rate of spread of head fires (Marsden-Smedley and Catchpole 1995b).

When planned burns are conducted, variations in fuel type and moisture within the burn area can be utilised to influence fire behaviour. In most sites, ridgelines and north-to-northwest facing slopes will have lower fuel moistures than gullies and south-to-southeast facing slopes. This results in ridgelines and north-to-northwest facing slopes typically having

more open vegetation, lower fuel loads and more frequent fires than gullies and south-to-southeast facing slopes. When planned burns are conducted where the fuel moisture in gullies and/or south-to-southeast facing slopes is too high to sustain burning, these areas can be used as control lines so that fires burn only ridgelines and/or north-to-northwest facing slopes.

The main types of control lines used are hand trails, tracks, roads, rivers, fuel-reduced areas and vegetation that is too wet to burn. Fires cross control lines mainly by direct flame (Photo 2) contact across the control line, spot fires, and to a lesser extent radiant heat igniting fuels on the unburnt side of the control line. Where planned burning is performed using narrow four-wheel-drive tracks or handlines, the upper fire intensity limit should be about 500 kW/m, or a flame height of about 1-2 m.

The most significant factors influencing the ability of fire crews to hold fire breaks are the length of fireline, ease of access, wind speed and fuel hazard (McCarthy *et al.* 2003). The last two of these factors are also major influences on the fire intensity and the potential for spot fires.
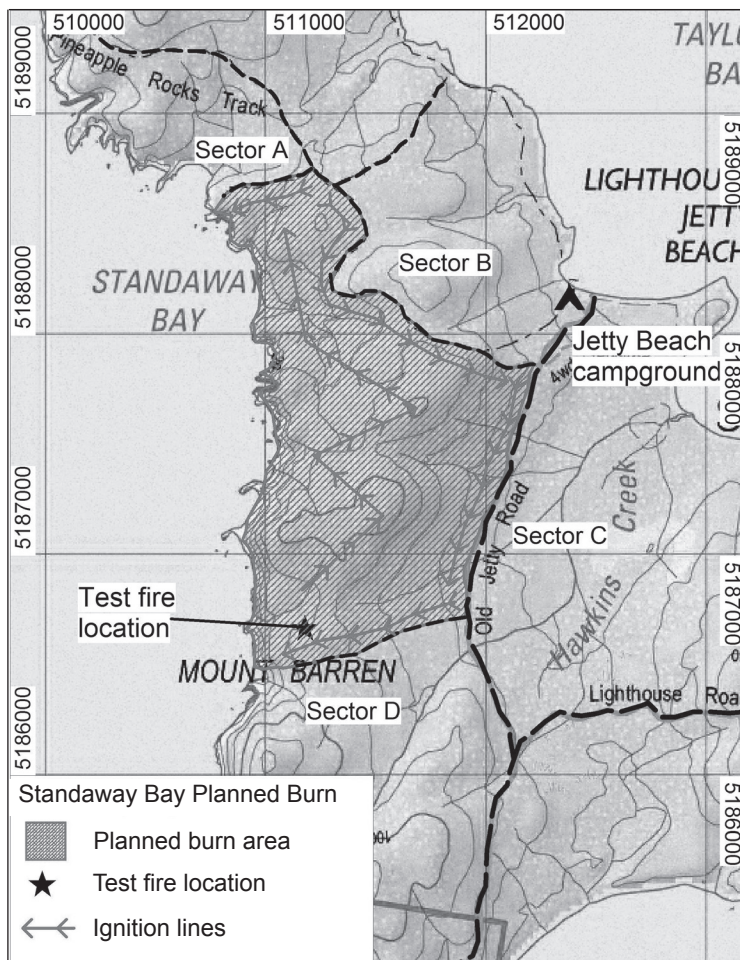

Figure 1. Map of planned burn at Standaway Bay, showing location of test fire, ignition lines and control lines. Grid squares are 1 km by 1 km.

Fire ecology, geomorphology, fire regime modelling and climate change

In Tasmania to date, and despite extensive debate regarding the interactions between fire and ecological and geomorphological factors, only limited research has been performed (Marsden-Smedley 2009; Brown 1993; Hannan *et al.* 1993; Jackson and Brown 1999; Mallick *et al.* 2007).

The impacts of fire on ecological values range between high-level, long-term adverse impacts, through short-term, low-level impacts, to the dependence on frequent fire for maintenance of species and structural diversity. The most dramatic example of high-level, long-term impacts is the effect of fire on western Tasmanian native conifers (in particular pencil pine, King Billy pine and Huon pine) and fagus (deciduous beech). These species are highly fire-sensitive, typically have cover and dominance greatly reduced by a single fire (often by > 99%), and take over 500 years to recover from a single fire (Gibson 1986; Brown 1988; Peterson 1990; Robertson and Duncan 1991; JB Kirkpatrick pers. comm.).

When managing for ecological values, a range of strategies can be applied for determining the most appropriate fire management regime. The following questions should be considered: is the aim to use a fire regime similar to that used by Aboriginals? is the aim to maintain the current regime? or, is the aim to develop a new regime based around plant and animal attributes?

The merits of and restrictions with using Aboriginal-style fire regimes in south-west Tasmania have been reviewed by Marsden-Smedley and Kirkpatrick (2000). The fire regime most likely utilised by Tasmanian Aboriginal people would have been frequent fires (e.g. on average less than about 20 years between fires), mostly of low-intensity, lit when scrub, eucalypt forest, rainforest and alpine areas were too wet to burn (Marsden-Smedley 1998a, 1998b;

Marsden-Smedley and Kirkpatrick 2000). This regime is analogous to the firestick farming regime proposed by Jones (1969), who found that such a fire regime, modified to meet contemporary requirements, had the potential to provide for appropriate management for ecological values.

All the Tasmanian vegetation associations suitable for planned burning have low fire sensitivity, high to very high flammability, and are ecologically adapted to recurrent fire. With a few exceptions, the vegetation types not suitable for planned burning have moderate to extreme fire sensitivity, low to moderate flammability, and in some cases fire results in marked reductions in species diversity (Pyrke and Marsden-Smedley 2005).

The degree to which a fire burns a site (and hence the proportion left unburnt) is of ecological concern, particularly for species that have to re-colonise post-fire from unburnt areas. With increasing time since fire, there is normally a corresponding increase in the proportion of the site subsequently burnt and a decrease in the size of unburnt patches. For example, during both the 2003 Arthur-Pieman and 2006 Reynolds Creek fires, <1% of the area of buttongrass moorland remained unburnt in areas last burnt >25 years previously, while in younger areas (areas last burnt <25 years previously) >50% remained unburnt (Parks and Wildlife Service wildfire and planned burn fire databases, unpublished).

Buttongrass moorland is the most comprehensively studied vegetation association from a Tasmanian fire ecology perspective (Jackson 1968, 1978; Mount 1979; Bowman 1980; Bowman and Jackson 1981; Brown and Podger 1982; Bowman *et al.* 1986; Jarman *et al.* 1988a, 1988b; Brown 1996, 1999; Jackson 1999; Jackson and Brown 1999; Brown *et al.* 2002; Mallick *et al.* 2007). Species diversity in buttongrass moorlands of low and medium productivity is highly resilient to changes in fire frequency and time since fire (Jarman *et al.* 1988a, 1988b;

Marsden-Smedley 1990; Brown *et al.* 2002), although frequent fire has greater effects at low-productivity sites than at medium-productivity sites. For example, observational data from low-productivity buttongrass moorlands in north-west Tasmania are consistent with the heath component of the moorland being reduced in its abundance by repeated frequent fires (JB Marsden-Smedley, unpub. data); in medium-productivity sites on the Navarre Plains frequent fire appears to have minor influence on species and structural diversity (JM Balmer, pers. comm.); and in higher productivity, low-altitude sites in northern Tasmania, buttongrass moorlands may be structurally transformed into a wet scrub association by the absence of fire for about 30 years (Marsden-Smedley and Williams 1993).

Structural factors may be more important than time since fire for fauna in buttongrass moorlands. Gellie (1980) considered that southern emu wrens, striated field wrens, swamp rats, broad-toothed mice and swamp antechinus require dense vegetation for cover and nesting, and that these species may take up to 15 years to recolonise areas following fires unless suitable pockets of unburnt vegetation are left as breeding areas. This is consistent with small mammals regaining their pre-fire densities once vegetation densities regain about 75% of their pre-fire levels (M. Driessen, pers. comm.). Arkell (1995) found a similar situation regarding small mammal diversity in buttongrass moorlands, with species diversity and number being highly correlated with moorland cover, but poorly correlated with time since fire. This means that the time period required for small mammal populations to recover following fires is the time period for cover values to reach about 65-75%, which varies from about four or five years in medium-productivity moorlands, to about 10-20 years in low-productivity moorlands (Plate 3).

Chaudhry *et al.* (2007) found that the critical factors controlling bird diversity in

moorlands appeared to be related to food availability and whether scrub boundaries and scrub along creek lines had been burnt, and not time since fire. This situation is similar to that found by Bryant (1991), who found that ground parrots in buttongrass moorland were common in sites more than about one year since fire, with peak densities at four to seven years since fire. The situation with orange-bellied parrots is more complex: this species may require feeding areas in buttongrass moorland to be burnt within the past three to 12 years (with older areas being unsuitable for feeding), with long-unburnt scrub and wet eucalypt forest being required for nesting (Brown and Wilson 1984).

The effect of time since fire on the abundance and diversity of invertebrates in buttongrass moorland was investigated by Greenslade and Driessen (1999), who found that both abundance and diversity were highest in sites of intermediate time since fire (11 to 19 years), with some evidence of declines in species diversity in sites more than 20 years after fire. The invertebrate species groups most strongly influenced by time since fire were mites, spiders, springtails, beetles, flies and moths. In contrast, Green (2007) suggests that mite diversity and abundance increase in buttongrass moorland unburnt for < 30 years. Mallick *et al.* (2007) also suggest that invertebrate species diversity of buttongrass moorland will be maintained in patches as small as 50 m by 50 m, while small mammals may require patches of up to one hectare. For burrowing crayfish in buttongrass moorlands, the critical issue for fire management is minimising fires under dry soil conditions (when SDI is < 50).

For dry eucalypt forests, some information on the ecological impact of planned burning is available from the south-eastern Australian mainland, which should be relevant to Tasmania. Frequent planned burning may increase species diversity in the understorey but decrease species diversity in the overstorey, with maximum

Photo 3. Buttongrass moorland five years after a planned burn.

species diversity being recorded at between one and five years after fire (Penman *et al.* 2008). Planned burning at three year intervals may have only minor impacts on birds, small mammals and invertebrates, with species diversity returning to pre-fire levels within four to five years post-fire, but burns more frequent than every 10 years may result in reductions in soil fertility and carbon (Department of Sustainability and Environment 2003). Differences may, however, occur between autumn and spring fires, with autumn fires being typically conducted at lower fuel moisture levels resulting in increased fine fuel removal, burning of logs and increased bark consumption compared to spring burns. The higher intensities and greater fuel removal in autumn burning may also result in greater regeneration of reseeding species, while spring burning may result in greater regeneration of resprouting species (Department of Sustainability and Environment 2003). In order to maximise ecological values, at least 40% of the site should be left unburnt in areas containing a mixture of fuel and vegetation types (e.g.

gullies, slopes and ridges), with at least 10% of each vegetation type being left unburnt. This means that an average fire frequency of 10 years between fires should be adequate to maintain species diversity, but 20 years between fires will be required to maintain structural diversity (Department of Sustainability and Environment 2003).

For threatened species in Tasmanian dry eucalypt forests, Bryant and Jackson (1999) recommend low-intensity mosaic burns at 8-14 year intervals for swift parrots, 10-14 year intervals for forty-spotted pardalotes, and 20-30 year intervals for velvet worms. In *Poa* grasslands burnt for management of the Ptunarra brown butterfly, Bryant and Jackson (1999; see also Bell 1999) recommended mosaic burns in autumn and winter at 4-7 year intervals, when the basal fuels in *Poa* tussocks are wet so that impacts to butterfly larva were minimised.

Fire has the potential to impact soil and geomorphological values in a number of ways. Fire can heat soils, cause changes to nutrient and/or carbon levels, expose

the soil surface to impacts from rain, increase surface flow rates and/or reduce soil infiltration rates (Department of Sustainability and Environment 2003; MacDonald and Huffman 2004). In addition, if soils have high organic contents (i.e. are organosols), then they may be directly impacted by being burnt in peat fires. Organosols are defined as having ≥ 20% soil organic matter where clay content is < 15%, or ≥ 30% soil organic matter if clay content is > 15% (Eggleton 2001).

Di Folco (2007) found that about 75% of buttongrass moorlands are underlain by mineral soils, not by organosols, and that the majority of sites that have organosols were located in very wet areas that rarely dry out. Di Folco (2007) also found that minimal erosion occurred in buttongrass moorlands following both dry-soil wildfires and wet-soil planned burns. As a result, the risk of soil erosion and degradation from planned burning in buttongrass moorlands conducted at low SDI is considered low (di Folco 2007, di Folco pers. comm., Storey 2008). The rate of organosol formation in Tasmania is currently unknown, but di Folco (2007) suggests that in buttongrass moorlands the rate of organosol formation is slow, with no significant changes being observed over a six year period.

A major challenge for fire management is predicting the long-term consequences of planned burning. The wildfire that is being pre-empted by fuel management burning may not occur for several decades, resulting in multiple planned burns being undertaken in the intervening time period. With ecological management burning, the challenge is to understand the potential impacts of multiple burns, performed in different seasons, frequencies, sizes, locations and/or intensities. Fire regime modelling has the potential to provide a long-term perspective on these issues (Cary 2002).

In Tasmania, fire regime modelling has only been conducted in southwest Tasmania

(King 2004a, 2004b; King *et al.* 2006, 2008). This modelling examined the effects of varying the amount of planned burning in buttongrass moorland, the burning strategy (i.e. broad-scale versus strategic burning), the size and distribution of burning blocks, the implications of climate change on the total area burnt, and (importantly) the area of fire-sensitive rainforest and alpine vegetation burnt. This modelling indicated that, regardless of the amount of planned burning conducted, the total area of buttongrass moorland burnt remained fairly constant. However, the area of fire-sensitive vegetation burnt decreased as the area of planned burning increased. This in turn suggests that planned burning has the potential to transform the fire regime from mostly high-intensity wildfires that burn all vegetation types, to mostly lower-intensity burns. The implications of climate change were also examined using fire regime modelling. In southwest Tasmania, modelling indicated that the projected changes in climate have the potential to increase the average annual area of rainforest and alpine areas burnt by about 38%.

Concluding remarks

This paper has reviewed background information and literature regarding planned burning. Subsequent papers (Marsden-Smedley and Whight 2011; Marsden-Smedley 2011) cover respectively fire risk assessment procedures for planned burning, and guidelines for conducting planned burning in Tasmania.

There are, however, some knowledge gaps and further research required in order to maximise the utility of planned burning in Tasmania. The usefulness of the Vesta fire model in south-eastern Australian dry eucalypt forests is being examined by a project being coordinated by the Department of Sustainability and Environment, Victoria, and the relationship between fire age, site conditions, forest type

and fuel hazard rating in Tasmanian dry eucalypt forests is currently the subject of a Tasmanian Fire Research Fund project. An improved general ecological knowledge of drier forests, including the dependence of species and structural diversity on fire age, fire frequency, season and intensity, is also required. Improved and/or enhanced fire behaviour models are required for heathland, dry scrub and wet scrub. Interactions between native animal grazing and fire potential in Tasmanian native grasslands are being researched in the School of Geography and Environmental Studies, University of Tasmania, but more data is needed on fire behaviour in these ecosystems. And, finally, more understanding is needed in regard to weed management with fire, particularly the prediction of fuel moisture. Information relevant to many of these areas could also be obtained opportunistically by collecting data from planned burns and wildfires.

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